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Reviewing the anaerobic digestion of food waste for biogas production

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ABSTRACT

The uncontrolled discharge of large amounts of food waste (FW) causes severe environmental pollution in many countries. Within different possible treatment routes, anaerobic digestion (AD) of FW into biogas, is a proven and effective solution for FW treatment and valorization. The present paper reviews the characteristics of FW, the principles of AD, the process parameters, and two approaches (pretreatment and co-digestion) for enhancing AD of food waste. Among the successive digestion reactions, hydrolysis is considered to be the rate-limiting step. To enhance the performance of AD, several physical, thermo-chemical, biological or combined pretreatments are reviewed. Moreover, a promising way for improving the performance of AD is the co-digestion of FW with other organic substrates, as confirmed by numerous studies, where a higher buffer capacity and an optimum nutrient balance enhance the biogas/methane yields of the co-digestion system.

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Contents

1. Introduction

1.1. Food waste (FW) generation

With the worldwide economic development and population growth, food waste is increasingly produced mainly by hotels, restaurants, families, canteens and companies. The amount of FW

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was nearly 90 million tons in China by 2010, as shown in Fig. 1. FW accounts for a large proportion in municipal solid wastes (MSW) in both of developed and developing countries, as shown in Tables 1 and 2 [1].

1.2. Characteristics of FW

According to the different eating habits, the FW composition will vary, with rice, vegetables, meat, eggs and other main components. As shown in Table 3, the total solid (TS) and volatile solid (VS) contents of FW were in the ranges of 18.1–30.9 and 17.1–26.35, respectively, indicating that water accounts for 70–80% in FW. Due to this high moisture content (MC), FW is an easily biodegradable organic substrate. Without any effective treatment measures, the disposal of FW has caused severe environmental pollution in many countries [2,3]. The traditional approaches for FW disposal were mainly landfill, incineration and aerobic composting. Whereas landfilling FW has been largely banned in many countries, incineration is energy-intensive (due to the high MC) and often creating air pollution. Both environmentally unfriendly approaches are gradually discarded. The application of FW as animal feed also bears a lot of risks since the propagation of diseases will be higher if FW is directly used as animal feed as a result of the shorter food chain. Laws are hence increasingly more severe with respect of environmental protection and to ensure food safety. Alternative methods for FW disposal are needed to tackle the waste crisis.

1.3. Principles of FW anaerobic digestion

As shown in Table 3, FW not only contains macromolecular organic matter, but also contains various trace elements. Currently, AD of FW has become an intensive field of research, since the organic matter in FW is suited for anaerobic microbial growth [6]. During the anaerobic process, organic waste is biologically degraded and converted into clean biogas [7]. According to Appels et al. [8], the biodegradation process mainly includes four steps: hydrolysis, acidogenesis, acetogenesis and methanogenesis, as shown in Fig. 2.

Differently, Molino et al. [9] pointed out that AD of organic waste can be split into three steps: hydrolysis, acidogenesis and methanogenesis.

No matter how many steps are involved in AD, the biodegradation processes of the both approaches are similar. Firstly, high molecular materials and granular organic substrates (e.g., lipids

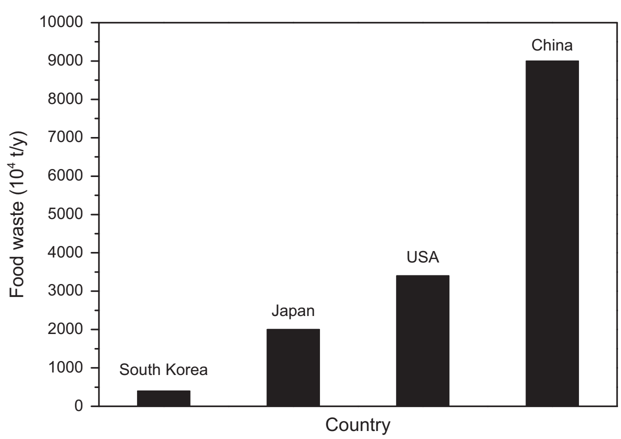

Fig. 1. The amount of FW discharged in some countries.

Table 1
Proportion of FW in MSW in China

Cities	Beijing	Shanghai	Guangzhou	Shenzhen	Nanjing	Shenyang
Percentage	37	59	57	57	45	62

Table 2
Proportion of FW in MSW in some countries

Countries	USA	England	France	Germany	Holland	Japan	South Korean	Singapore
Percentage	12	27	22	15	21	23	23	30

Table 3
Characteristics of FW reported in literatures

Parameters	Zhang et al. [2]	Zhang et al. [3]	Zhang et al. [4]	Li et al. [5]
TS (%, w.b.)	18.1 (0.6)	23.1 (0.3)	30.90 (0.07)	24
VS (%, w.b.)	17.1 (0.6)	21.0 (0.3)	26.35 (0.14)	232
VS/TS (%)	0.94 (0.01)	90.9 (0.2)	85.30 (0.65)	94.1
pH	6.5 (0.2)	4.2 (0.2)	–	–
Carbohydrate (%, d.b.)	61.9	–	–	55.2
Protein (%, d.b.)		–	–	15
Fat (%, d.b.)	23.3 (0.45)	–	–	23.9
Oil (%, d.b.)	–	4.6 (0.5)	–	–
C (%, d.b.)	46.67	56.3 (1.1)	46.78 (1.15)	54
N (%, d.b.)	3.54	2.3 (0.3)	3.16 (0.22)	2.4
C/N	13.2	24.5 (1.1)	14.8	22.5
S (ppm, w.b.)	0.33	–	2508 (87)	8.6
P (ppm, w.b.)	1.49 (0.09)	–	–	88
Na (%, d.b.)	0.84	3.45 (0.2)	–	2.24
K (%, d.b.)	0.3	2.30 (0.04)	0.90 (0.11)	–
Ca (%, d.b.)	0.07	0.4 (0.01)	2.16 (0.29)	2.44
Mg (%, d.b.)	0.03	0.16 (0.01)	0.14 (0.01)	–
Fe (ppm, w.b.)	3.17	100 (23)	766 (402)	–
Cu (ppm, w.b.)	3.06	–	31 (1)	–
Zn (ppm, w.b.)	8.27	160 (30)	76 (22)	–
Al (ppm, w.b.)	4.31	–	1202 (396)	–
Mn (ppm, w.b.)	0.96	110 (95)	60 (30)	–
Cr (ppm, w.b.)	0.17	–	<1	–
Ni (ppm, w.b.)	0.19	–	2 (1)	–

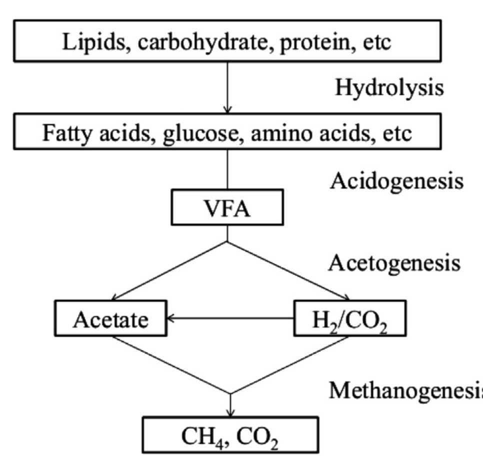

Fig. 2. Four steps in the AD of organic substrates.

and carbohydrates, protein) are hydrolyzed by fermentative bacteria into small molecular materials and soluble organic substrates (e.g., fatty acids and glucose, amino acids): generally, hydrolysis is regarded as the rate-limiting step in the AD of solid organic wastes, because the hydrolytic enzyme should be primarily adsorbed on the surface of solid substrates [10,11]. Secondly, small molecular materials and granular organic substrates are degraded into volatile fatty acids (e.g., acetate, propionate and butyrate) along with the generation of by-products (e.g., NH_3, CO_2 and H_2S). Thirdly, the organic substrates produced in the second step are further digested into acetate, H_2, CO_2 and so on, which could be used by methanogens for methane production.

Generally, the substrates which could be utilized by methanogens include three types: (1) short-chain fatty acid (C1–C6); (2) n- or i-alcohols; (3) gases (CO, CO_2 and H_2). According to Appels et al. [8], methane could be obtained by two groups of methanogens: one group mainly uses acetate for methane production, and the other group mainly uses H_2/CO_2. In addition to acetate and H_2/CO_2, formate, carbinol and CO could also be transformed into methane as shown in Fig. 3 [12]. During the methane formation process, the co-enzymes M and F_{420} play a significant role in formate and CO transformation. Formate and CO are firstly transformed into CO_2 by F420, and then CO_2 is reduced into CH_4 along with the action of co-enzyme M. Moreover, the co-enzyme M also plays an important role in acetate and carbinol transformation. Since VFA is the main product during anaerobic digestion, most of the methane is produced through the acetate route [8,10].

Currently, most of anaerobic digesters are single-stage systems, which e.g. accounts for 95% of the European full-scale plants [13]. It should however be remembered that the different digestion systems dealing with high solid content feed stocks, have their specific operating conditions and characteristics, as summarized in Table 4.

AD of FW is a complex process that should simultaneously digest all organic substrates (e.g., carbohydrate and protein) in a single-stage system. It is governed by different key parameters such as temperature, VFA, pH, ammonia, nutrients, trace elements, and others: a good nutrient and trace element balance, and a stable environment are required for microbial growth. It is therefore extremely important to maintain the key parameters within the appropriate range for long term operation of AD. These key parameters are discussed in the following section.

2. Key parameters

2.1. Temperature

Temperature is one of the most significant parameters influencing AD, because it not only influences the activity of enzymes and co-enzymes, but also influences the methane yield and digestate (effluent) quality [8,15]. Generally, anaerobic bacteria can grow at psychrophilic (10–30 °C), mesophilic (30–40 °C) and thermophilic (50–60 °C) conditions [16,17]. The performance of AD however increases with an increasing temperature [15,17], stressing the advantages of the thermophilic operation with its higher metabolic rates, higher specific growth rates, and higher rates of the destruction of pathogens along with higher biogas production [17,18]. Gallert et al. [19,20] verified that thermophilic digestion suffers less from the inhibition by ammonia accumulation than mesophilic digestion. Wei et al. [21] pointed out that the biogas production under thermophilic (55 °C) conditions was more than double the output under psychrophilic (15 °C) condition. Moreover, the rates of organic nitrogen degradation and phosphorus assimilation also increased with temperature [15]. Thermodynamics show that a higher temperature is a benefit to endergonic reactions (e.g., the breakdown of propionate into acetate, CO_2, H_2), but not favorable to exergonic reactions such as hydrogenotrophic reactions and methanogenesis [8,22]. Furthermore, temperature could also affect the passive separation of solids which is demonstrated to be better under thermophilic than psychrophilic conditions [23].

Although several advantages were observed under thermophilic condition, some disadvantages are worth considering since the

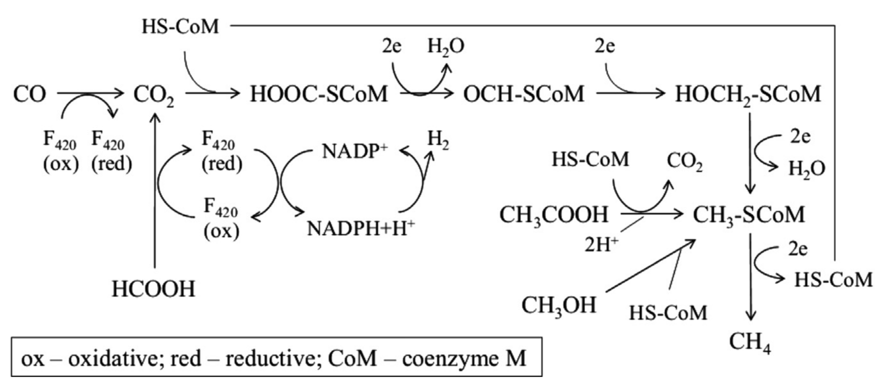

Fig. 3. Metabolic process from CO, CO_2, formate, carbinol and acetate to methane.

Table 4
Comparison of different digester configurations for high solid content feedstock [14]

Criteria	One-stage versus two-stage AD		Batch versus continuous AD	
	One-stage	Two-stage	Batch	Continuous
Biogas production	Irregular and discontinuous	Higher and stable	Irregular and discontinuous	Higher and stable
Solid content (%)	10–40	2–40	25–40	2–15
Relative cost	Low	High	Low	High
Volatile solids destruction	Low to high	High	40–70%	40–75%
HRT (days)	10–60	10–15	30–60	30–60
OLR (kg VS/m³ day)	0.7–15	10–15 in second stage	12–15	0.7–1.4

thermophilic process is more sensitive to environmental changes than the mesophilic process [18,24–26]. Process failure can be obtained when the rate of temperature changes excesses 1 °C/day, and the changes in temperature should be less than 0.6 °C/day to maintain a stable digestion [8]. Moreover, recent research found that the overall rate of solubilization of FW was significantly lower under thermophilic (55 °C and 65 °C) than under mesophilic conditions [27].

2.2. VFA and pH

Volatile fatty acids (VFAs) which mainly include acetic acid, propionic acid, butyric acid, and valeric acid, are the main intermediate products during AD of organic wastes [28–31]. Generally, VFAs produced in the anaerobic process could be ultimately transformed into CH_4 and CO_2 by syntrophic acetogens and methanogenic bacteria. However, VFAs can be accumulated at high organic loading, resulting in the decrease of pH and even the failure of AD [28,31,32]. Among the four acids, acetic and propionic acids play a dominant role in biogas production [3], and their concentrations could be used as indicator of the performance of AD [28]. Previous research demonstrated that a propionic acid to acetic acid ratio exceeding 1.4 or the concentration of acetic acid exceeding 0.8 g/L lead to AD failure [28,33,34]. The propionic acid to acetic acid ratio could hence be utilized as the indicator of digestion imbalance [28,35].

Several conventional methods for monitoring VFAs, such as ion-exclusion, high performance liquid chromatography (HPLC) and gas-chromatography (GC), are used with simple pretreatment. However, these methods are usually hysteretic, time-consuming, and these material-demanding analyses are not reliable for field-work applications [31]. Recently, on-line methods based either on GC or on titration and back titration methods, were developed to overcome the defects of traditional methods, and were proven to be available [28,31,36,37]. Based on on-line methods, analysis of the performance of AD could be carried out real-time, and measures could be taken in time to avoid digestion imbalance.

VFAs determine the pH which is also one of the most important parameters affecting AD. Anaerobic bacteria need different pH ranges for their growth, e.g., a comprehensive pH range of 4.0–8.5 is required by fermentative bacteria while a limiting range of 6.5–7.2 is favorable for methanogens' growth [8,38,39]. Previous reports pointed out that the VFAs could be significantly affected by the pH of anaerobic digester: at low pH the main VFAs are acetic and butyric acids, while acetic and propionic acid played a dominant role when pH was 8.0 [8,40]. Moreover, both of the type of acid-producing bacteria and the bacteria number could be controlled by the pH control [41,42].

2.3. C/N ratio

The performance of AD is significantly affected by C/N ratio [43–45]. An optimum C/N ratio is needed for AD because an appropriate nutrient balance is required by anaerobic bacteria for their growth as well as for maintaining a stable environment. Generally, a C/N ratio range of 20–30 was considered to be the optimum condition for AD [46,47]. Anaerobic co-digestion of three organic substrates (dairy, chicken manure and wheat straw) was performed by Wang et al. [48], who explained that the maximum methane potential was achieved at the C/N ratio of 27.2, with stable pH and low concentrations of total ammonium-N and free NH_3. Similar results were reported by Zeshan et al. [43] who found that the AD performed well at a C/N ratio of 27 than 32. However, recent studies pointed out that digestion proceeded well at low C/N ratios (15–20). By co-composting green waste and FW, Kumar et al. [44] proved that organic substrates could be digested effectively at a C/N ratio of 19.6. Zhong et al. [45] also found that the optimal C/N ratio for the co-digestion was 20. Zhang et al. [32] pointed out that the optimum C/N ratio was 15.8 when co-digestion FW with cattle manure (CM). The above findings indicated that the optimum C/N ratio for AD depends on both of the feedstock and the inoculum. No matter what the C/N ratio is, an appropriate balance between C and N is required for effective digestion. An optimum carbon content had a positive effect on avoiding excessive ammonia inhibition [45,49,50]. Recent studies found that VFA could form buffer system with ammonia, resulting in higher methane yield and TOC utilization [51]. Therefore, an adjustment of the C/N ratio is needed for stable AD in a long-term operation.

2.4. Ammonia

Ammonia is formed during the biodegradation process of protein or other nitrogen-rich organic substrates, and mainly exists in the form of ammonium (NH_4^+) and free ammonia (NH_3) [8,52,53]. It could be utilized as an essential nutrient for bacterial growth though it can also be toxic to microbes in the presence of high concentrations [7,52,54]. It is well known that ammonia plays an important role in balancing the C/N ratio which could affect the performance of AD significantly [48]. In general, the ammonia was at a low level when the C/N ratio of the feedstock (e.g., crop residues) was beyond 30, resulting in lower performance of AD as well as lower methane yield. Many previous papers have reported that ammonia could enhance the buffer capacity of the AD, because VFAs formed during digestion process could be neutralized by ammonia [3,48,51,55]. The reaction equations (Eqs. (1)–(3)) between ammonia and VFAs have been reported by Zhang et al. [3], and are as follows:

$$C_xH_yCOOH \rightleftharpoons C_xH_yCOO^- + H^+ \quad (1)$$

$$NH_3 \cdot H_2O \rightleftharpoons NH_4^+ + OH^- \quad (2)$$

$$C_xH_yCOOH + NH_3 \times H_2O \rightarrow C_xH_yCOO^- + NH_4^+ + H_2O \quad (3)$$

where C_xH_yCOOH represent the VFAs. VFAs accumulation will be observed when the organic loading rate (OLR) increases, leading to a risk of digestion failure. The ammonia however could react with these VFAs, avoiding the inhibition from VFAs and allowing enough VFAs for biogas production.

Despite its buffer capacity, ammonia was proven to be an inhibitor to lots of bacteria at high concentrations [8,43,56,57], since free ammonia can diffuse the cell membrane and further hinder cell functioning through disrupting the potassium and proton balance inside the cell [58]. Many previous reports pointed out that the sensitivity to ammonia of acetoclastic methanogens, which convert acetate into CH_4 and CO_2, are much higher than the hydrogenotrophic methanogens [54,56], and thus more likely to cease methane production [59]. Amongst methanogens, *Methanosaeta concilii* and *Methanosarcina barkeri* showed higher sensitivity to increasing free ammonia concentrations [60]. The free ammonia concentration increases with increasing temperature and pH value, e.g., for the condition at pH 7 and 35 °C, less than 1% of the total ammonia is in the form of free ammonia. However, at the same temperature, the free ammonia increases to 10% at pH 8, as shown in Fig. 4 [57].

Low concentrations of ammonia are needed for bacteria growth, whilst higher levels result in inhibition of microorganisms [8,59]. A wide range of critical concentrations initiating ammonia inhibition was reported previously. Chen et al. [56] summarized the critical concentrations and pointed out that a 50% reduction in the methane yield will occur as a result of the ammonia concentration of 1.7–14 g/L. A review by Yenigün and Demirel [52] also

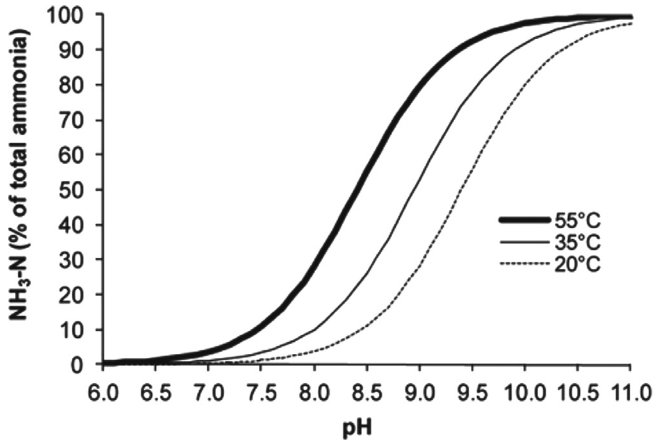

Fig. 4. Free ammonia and ammonium percentages present in solution at 20, 35 and 55 °C and varying pH.

stated that digestion failure could be caused by the ammonia concentration of 1.7–1.8 g/L (free NH_4^+) and specifically illustrated that the inhibition was due to free ammonia rather than the ammonium ions. The wide range of ammonia concentration for causing inhibition depends on the differences in feedstock, inoculums, environmental conditions (e.g., temperature and pH) [52] and acclimatization periods [56].

High concentration of ammonia could not only lead to lower biogas production and even digestion failure, but also result in ammonia emission from effluent [61]. To promote the performance of AD, various approaches for ammonia removal has been well studied by many researchers [54,61–64], including ion exchange [65], ammonia stripping [54,63,66], biological nitrogen elimination processes [67], electrochemical conversion [68], microwave [69], and membrane contactors [59].

2.5. Long chain fatty acids

FW is a lipid-rich resource in which the lipid concentration is about 5.0 g/L [3,70]. Long chain fatty acids (LCFA) are mainly composed of oleic acid (C18:1), linoleic acid (C18:2) and palmitoleic acid (C16:0), being the main intermediate by-products of the lipid degradation process [71,72]. LCFA can be further converted to hydrogen and acetate by acetogenic bacteria through a β-oxidation process, and finally to methane by methanogenic archaea [71,73, 74]. However, the LCFA biodegradation process is considered to be the rate-limiting step of AD process [75,76]. Some previous researchers attributed the rate-limiting to the initial concentrations of LCFAs: higher concentration of LCFA result in failure of AD [75,77]. Others explained that microbial flocs floating on the surface and LCFAs inhibition of anaerobic microorganisms: LCFAs adsorption onto the cell wall and membrane which affects the metabolic processes of transportation [78–80]. Researchers found that inhibition of LCFA depends on the bacterial groups and also on the different acids. Lauric and myristic acids are the strongest inhibitors of different bacterial groups [76]. The inhibition intensity from unsaturated fatty acids increases with the number of double bonds and the chain size. Moreover, the degree of inhibition by a mixture of fatty acids was greater than the individual effect of each acid [81].

Serious process problems in biogas plants are usually observed due to the inhibition of LCFA, because the inhibition could be caused at lower concentrations, with IC50 values at concentrations of 50–75 mg/L for oleate [82], 1100 mg/L for palmitate [83], 1500 mg/L for stearate [84], at mesophilic condition. However, the potential for methane production from LCFA was substantial. Theoretically, the methane potential of lipids is 1014 L/kg-VS, a value obviously higher than of carbohydrates (e.g., 370 L/kg VS for glucose) [74,85]. To overcome the inhibition from LCFA, strategies for recovering inhibition caused by long chain fatty acids have been studied by many researchers. Palatsi et al. [80] found that increasing the biomass/LCFA ratio through dilution with active inoculum, and adsorbing the LCFA and reducing the bio-available LCFA concentration through addition of adsorbents were effective recovery approaches. Zhang et al. [32] explained that anaerobic co-digestion of FW with CM could enhance the biodegradation process, resulting in a higher methane yield. Similar results have also been reported by others [86]. To enhance the activity of the anaerobic community and the efficient transforming of lipid-rich effluents, discontinuous feeding was also recommended [74].

2.6. Metal elements

Besides nutrient elements (C, H, O, N), metal elements including light metal ions (Na, K, Mg, Ca, Al) and heavy metal ions (Cr, Co, Cu, Zn, Ni, etc.) [87], are also required by anaerobic bacteria because these cations play an important role in enzyme synthesis as well as maintaining enzyme activities [2,88–90]. However, inhibition could be caused by both of light and heavy metal elements when their concentrations are too high [8].

Chen et al. [56] reported that the optimum concentration of sodium for mesophilic hydrogenotrophic methanogens was 350 mg/L, and at the concentration of less 400 mg/L, potassium could enhance the performance of both of thermophilic and mesophilic AD. Jackson-Moss et al. [91] reported that no inhibition on AD was observed when the concentration of calcium increase to 7000 mg/L. However, Yu et al. [92] pointed out that the optimum concentration for calcium was 150–300 mg/L. The lower optimum concentration of calcium was also reported in other literature. Huang and Pinder [93] found that a concentration higher than 120 mg/L could lead to an inhibition of cellular metabolism in the biofilm system. Kugelman and McCarty [94] reported a toxicity threshold concentration of 200 mg/L for calcium. Moderate inhibition occurred at the concentration of 2500–4000 mg/L, and strong inhibition occurred at a concentration of 8000 mg/L. A recent study found that addition of calcium could promote the formation of calcium stearate in AD., resulting in lower concentrations of LCFAs and higher performance of AD [95].

Unlike many other toxic substances, heavy metals are not biodegradable and can accumulate to inhibition concentrations [56,96]. Heavy metal elements could cause inhibition to anaerobic organisms due to the disruption of the enzyme function and structure [56]. Many previous findings have pointed out that the inhibition degree depends upon many factors, such as the total metal concentration, chemical forms of the metals, pH, and redox potential [97,98]. The inhibition concentrations of various metal elements have also been reported previously [8,56].

Inhibition of heavy metal elements was usually observed when AD of municipal sewage and sludge or industrial wastewater [8,56]. Differently, the concentrations of heavy metal elements in FW were always insufficient [2,6], while the light metal elements such as sodium and potassium were generally at high concentrations [3,99], as shown in Table 3. To enhance the performance of AD, addition of metal elements or metal-rich organic substrates has been studied by many researchers. Anaerobic co-digestion of FW with piggery wastewater was studied by Zhang et al. [2]. Results from co-digestion illustrated that trace elements played an important role in improving the performance of AD. Similar results were obtained by Zhang et al. [32] who found the trace elements played an important role in the performance enhancement during the anaerobic co-digestion of FW with CM. Zhang and Jahng [6] found that the stabilization of anaerobic system with trace elements addition was obviously enhanced in comparison of the system which FW was digested alone. Banks et al. [100] reported that selenium is essential for both propionate oxidation and syntrophic hydrogenotrophic methanogenesis. Selenium supplementation allows digestion to proceed at

substantially higher organic loading rates. To avoid the inhibition from sodium and potassium, AD in a dual solid–liquid (ADSL) system was proposed by Zhang et al. [3], who found that most of the sodium and potassium existed in the liquid fraction. By the application of the ADSL system, the methane yield was obviously improved in semi-continuous digestion. The above literature indicated that addition of metal elements or metal-rich organic substrates is an effective approach for improving the performance of AD, and measures should be taken to avoid the inhibition from sodium and potassium.

3. Pretreatments

AD of FW was significantly affected by the mass transfer in each biological step, as well as by food availability [101]. As discussed in Section 1, the hydrolysis is the rate-limiting step during the whole anaerobic process. Therefore, it is important to enhance hydrolysis for improving the performance of AD. Many factors including particle size, structure, component of feedstock, can influence the rate of hydrolysis, especially during the biodegradation of high molecular compound and granular substrates. To accelerate the hydrolysis step, several approaches for organic substrates pretreatment have been proposed, as shown in Table 5. The most common disintegration methods were mechanical grinding [102], ultrasound [103,104], microwave [105], thermal, chemical [106] or their combination [107,108], and biological pretreatment [109].

FW residues mainly comprise of carbohydrate polymers (starch, cellulose, hemicellulose), lignin, other organics (proteins, lipids, acids, etc.) and a remaining, smaller inorganic part [107]. Most of the carbohydrate polymers and protein exist in the solid form, such as rice, vegetables and meat. Physical pretreatment including milling and grinding, is important for improving the performance of AD because the particle size of these solid materials has significant effect on the rate of hydrolysis. Kim et al. [110] pointed out that the rate coefficient of the maximum substrate utilization doubled when the average particle size of FW decreased from 2.14 mm to 1.02 mm. Izumi et al. [101] also found that the particle size could significantly influence AD of FW. Approximately 40% improvement on the total chemical oxygen demand (total COD) was observed when FW was pretreated by a bead mill along with the mean particle size decreased from 0.843 mm to 0.391 mm, resulting in a higher methane yield. In addition, Izumi et al. further proved that excessive reduction of the particle size of the FW was not in favor of methane production because smaller particle substrates accelerated VFA accumulation. According to their research, the maximum cumulative methane production was obtained at the particle size of 0.6 mm.

Physical pretreatment was mainly focused on the particle size of substrates. By contrast, more attention was paid to the reactions during hydrolysis step in chemical pretreatment. Hydrolysis of cellulosic material and starch components can improve the rates of subsequent enzymatic reactions, increase the yield of sugar, and lead to the breakage of glycoside bonds of polysaccharides appearing in oligosaccharides, maltodextrins and monosaccharides [107]. Vavouraki et al. [107] pointed out that chemical pretreatment (using either 1.12% HCl for 94 min or 1.17% HCl for 86 min (at 100 °C)) on FW could increase the concentration of

Table 5
Pretreatment approaches of organic substrates degradation.

Pretreatment methods	Measures		Raw material(s)	Results	Mechanism	Reference
Physical	Mechanical grinding	Pretreatment at 30 bar	Waste activated sludge (WAS)	Shorted the sludge retention time from 13 to 6 days.	Increased soluble organic substrates	[102]
	Ultrasound	With frequency of 40 kHz power of 500 W	Municipal sludge	Resulted in VS reduction up to 47% and higher biogas yield	Enhanced sludge solubilization though cell lysis	[104]
	Microwave	145 °C	FW	Increased biogas production	Disrupted sludge and increased solubilization	[105]
	Thermal	120 °C + 30 min	FW	Biogas production increased by 11%	Increasing the solubilization	[106]
	Freezing–thaw	−80–55 °C	FW	Biogas production increased by 23%	Cell disruption	[106]
	Pressure–depressure	Pressure changed from 10 bar to 1 bar with CO_2 as pressurizing gas	FW	Biogas production increased by 35%	Breaking up the microbial cell walls and increasing the solubilization	[106]
Chemical	Acid	With 10 mol/L HCl at room temperature (18±2 °C) until pH 2 for 24 h	FW	Biogas production decreased by 66%	Forming inhibitors	[106]
Biological	Biological solubilization	FW+water	FW	Decreased organic concentration in the effluent	Increasing the solubilization	[109]
Physical–Chemical	Thermo-acid	With 10 mol/L HCl at room temperature (18±2 °C) until pH 2 for 24 h, and then 120 °C + 30 min	FW	Biogas production increased by 18%	Increasing the solubilization	[106]
	Thermo-acid	1.12% HCl for 94 min or 1.17% HCl for 86 min (at 100 °C)	MSWs	Soluble sugars increased by 120%	Increasing the solubilization	[107]
	Biological–physicochemical	Bacillusat 9 wt%, ultrasonic for 10 min and 500 mg/L citric acid.	Oily wastewater	Biogas production increased by 280%	Enhanced the oil degradation	[131]

Table 6
Co-digestion of FW with other organic substrates for improving performance of anaerobic process

Feedstock	Action of co-digestion	Influencing factor	Reference
FW+CM	Improve methane yield and system stability	High buffering capacity and trace elements supplement	[32]
FW+CM	Improve biogas production	High buffering capacity from ammonia	[103]
FW+CM	Improve methane yield	Nutrient balance	[121]
FW+CM	Improve methane yield	Nutrient balance	[122]
FW+CM	Increase energy returns and reduce GHG emissions	Nutrient balance	[123]
FW+livestock waste	Improve methane yield and VS reduction	Higher buffering capacity	[7]
FW+CM+oil	Improve methane yield	High buffering capacity	[124]
FW+CM+fat	Improve methane yield	lipids supplement	[120]
FW+CM+card packaging	Allow higher organic loadings and gave a more stable process	Trace elements supplement	[125]
FW+animal slurry wastewater	Improve both methane yield and TOC utilization	High buffering capacity	[51]
FW+piggery wastewater	Improve biogas productivity and process stability	Trace elements supplement	[2]
FW+yard waste	Improve methane yield	Less VFA accumulation	[126]
FW+distiller's grains	Increase biogas production	High buffering capacity from ammonia	[127]
FW+dewatered sludge	Enhance system stability	Less inhibition from Na+	[99]
FW+sewage sludge	Afford high organic loading rate	High buffering capacity from ammonia	[128]
FW+green waste	Improve VS reduction	C/N ratio	[44]
FW+press water	Improve methane yield and system stability	High buffering capacity	[129]
FW+brown water	Improve methane yield	High buffering capacity	[130]

soluble sugars by 120% in comparison of untreated FW. Similar chemical pretreatment in combination with ultrasonic and steam pretreatment for decomposing glycosidic bonds in starch was also reported previously [111,112]. Ma et al. [106] compared five different approaches (pressure–depressure, freeze–thaw, acid, thermo-acid, thermo) for FW pretreatment before AD, and found that the highest cumulative biogas production was obtained with the pressure–depressure method.

Although biogas/methane production was obtained, some of the pretreatment approaches had their disadvantages. For example, carboxylic acids, furans and phenolic compounds could be possibly formed in the acid pretreatment, resulting in inhibition to AD and less biogas production [113,114]. The cell membranes could be disintegrated in thermal pretreatment, as a result, the solubilization could be improved; however, limitation on the biodegradation of the hydrolysates will also be enhanced [106,115]. Moreover, pretreatments usually result in higher capital costs because of the additional energy or chemicals required [116,117]. The additional methane produced as a result of pretreatments was, in some cases, insufficient to offset the additional costs [118,119], and thus resulting in the pretreatment unfeasible financially [117].

Although a higher biogas/methane production was obtained, some of the pretreatment approaches have their disadvantages. For example, carboxylic acids, furans and phenolic compounds can be possibly formed in the acid pretreatment, resulting in inhibition to AD and less biogas production [113,114]. The cell membranes could be disintegrated in thermal pretreatment: as a result, the solubilization could be improved, but the limitation of the biodegradation of the hydrolysates will also be enhanced [106,115]. Moreover, pretreatments usually result in higher capital costs because of the additional energy or chemicals required [116,117]. The additional methane produced as a result of pretreatments was, in some cases, insufficient to offset the additional costs [118,119], and thus resulted in a negative economy of the combined process [117].

4. Anaerobic co-digestion

FW is a promising organic substrate for AD due to its high potential for methane production [120]. However, inhibition always occurred when FW is digested alone in the long-term operation. The reasons for the inhibition are that the nutrients always imbalance in the anaerobic digester: whilst e.g., trace elements (Zn, Fe, Mo, etc.) are insufficient, macronutrients (Na, K, etc.) are excessive [2,32,121], and the C/N ratio of FW was outside of the optimum reported in literature [3,14]. In addition, the concentration of lipids in FW is always higher that the limited concentration, which leads to inhibition [3,70]. To counteract the inhibition and to overcome the disadvantages in single digestion, co-digestion of FW with other organic substrates such as CM, wastewater, sewage sludge and green waste, had been widely carried out, as shown in Table 6.

Zhang et al. [32] found that co-digestion of FW with CM did not only improve the maximum acceptable organic loading rates (10 g-VS/L–15 g-VS/L), but also promoted the methane yield in semi-continuous digestion. A high buffer capacity was observed in co-digestion systems due to the improved ammonia concentration in CM. In addition, the biodegradation of lipids increased, resulting in a higher methane yield. Li et al. [122] obtained a 44% improvement on the methane yield by co-digestion of FW with CM. They also verified that the acids produced during AD play an important role in pre-treating the fibers in CM, resulting in a higher methane yield. Similarly, co-digestion of FW with CM to improve biogas production and methane yield has also been reported by other researchers [103,121–123]. Co-digestion of FW with CM balances the nutrients in the anaerobic digester, and thus provides a more stable environment for anaerobic bacteria. Neves et al. [120,124] pointed out that higher methane yield could be obtained through adding lipids into the co-digestion system, because of the high potential for methane yield of lipids and the higher biodegradation of lipids in co-digestion system. Moreover, anaerobic co-digestion of FW with other organic wastes can also improve biogas production and methane yield, as shown in Table 6. The main factors improving the performance of AD are due to the higher buffer capacity caused by higher ammonia from the organic wastes, the optimum C/N ratio in anaerobic digester, and the trace elements supplement. These references shown in Table 4 illustrate that anaerobic co-digestion of FW with one or more substrates is a promising approach for biogas and methane yield improvement, and this approach could be applied in pilot-scale and full-scale plants for a richer energy gas.

5. Perspectives and recommendations for the anaerobic digestion of food waste

FW is considered to be one of the most promising energy sources for renewable energy production, provided AD pretreatment is specifically adapted to this type of waste. By pretreatment, the nutrient concentration of FW can be readjusted to improve the performance of AD, by e.g. reducing the concentration of lipids, resulting in higher biogas production and lower lipids limitation [3]. Moreover, the

nutrient imbalance of FW is overcome through co-digestion of FW with other biomass waste (e.g., cattle manure, whey, municipal wastewater), resulting in a more appropriate C/N ratio and metal elements concentration for AD. Using a life cycle assessment, Evangelisti et al. [132] investigated three different but possible treatment technologies for organic fractions of MSW in the London area: (i) landfill with electricity production; (ii) incineration with steam recovery for combined heat and power (CHP) and (iii) anaerobic digestion with energy recovery as CHP. The life cycle inventory data revealed that AD emerges as the best treatment option for organic wastes. Similar results were also obtained by Cherubini et al. [133,134]. The previous results illustrated that AD is a reliable way for bio-energy recovery from FW.

In the long-term development perspective, additional research is needed. Firstly, the economic performance of AD of FW should be improved, and higher methane contents in the produced biogas will facilitate its co-use with natural gas [133]. Too low a methane content increases the cost of the biogas upgrading process. To improve the economics of biomass utilization, Budzianowski [135] proposed new concepts such as "negative net CO_2 emissions" in which biogas could be converted into hydrogen. In addition, the recommendations on design of biogas-based energy systems were of Budzianowski [136] are important to secure the future of AD of FW: (i) apply a distributed production of biogas to avoid costs of long-distance transportation of high-moisture content biomass, and (ii) centralize large scale decarbonized biogas-to-electricity power plants. Secondly, a highly perfected FW management system should be established by the government to centralize FW for large scale AD plants since FW is produced by different large and small restaurants, companies, schools and families at distinct and non-centralized places. To encourage people to collect their FW, the Chinese government has e.g. issued a new regulation: non-residential buildings are required to pay $4/ton for disposing of FW and $12/ton for other refuse after 2012 [137].

6. Conclusions

AD is a reliable technology for recycling bio-energy from FW. FW could be digested effectively under both mesophilic and thermophilic conditions. A buffer system could be formed by VFA and ammonia, resulting in higher methane yield and system stability. Trace element supplement to FW are favor the AD of FW since the trace elements in FW are usually insufficient. The concentration of lipids is usually higher than the limit concentration, resulting in inhibition of AD. However, lipids are high potential bio-resources for methane production. Co-digestion of FW with other substrates such as CM could enhance the biodegradation of LCFA as well as the methane yield. In addition, co-digestion could also improve the buffer capacity and result in increased acceptable organic loadings in comparison with single digestion.

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